

Future carbon storage in harvested wood products from Ontario's Crown forests

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Abstract: This analysis quantifies projected carbon (C) storage in harvested wood products (HWP) from Ontario's Crown forests. The large-scale forest C budget model, FORCARB-ON, was applied to estimate HWP C stock changes using the production approach defined by the Intergovernmental Panel on Climate Change. Harvested wood volume was converted to C mass and allocated to four HWP end-use categories: in use, landfill, energy, and emission. The redistribution of C over time among HWP end-use categories was calculated using a product age-based C-distribution matrix. Carbon emissions for harvest, transport, and manufacturing, as well as emission reductions from the use of wood in place of other construction materials and fossil fuels were not accounted for. Considering the wood harvested from Ontario Crown forests from 1951 to 2000 and the projected harvest from 2001 to 2100, C storage in HWP in use and in landfills is projected to increase by 3.6 Mt-year⁻¹ during 2001–2100, with an additional 1.2 Mt-year⁻¹ burned for energy. Annual additions of C projected for HWP far outweighs the annual increase of C storage in Ontario's Crown forests managed for harvest, which is projected to increase by 0.1 Mt-year⁻¹ during the same period. These projections indicate that regulated harvest in Ontario results in a steadily increasing C sink in HWP and forests. Uncertainties in HWP C estimation are also discussed.

Résumé : Cette analyse quantifie la séquestration prévue du carbone (C) dans les produits du bois provenant des forêts publiques de l'Ontario. Le modèle de bilan à grande échelle de C de la forêt, FORCARB-ON, a été utilisé pour estimer les changements dans les stocks de carbone associés aux produits du bois à l'aide de l'approche de production définie par le « Intergovernmental Panel on Climate Change ». Le bois récolté a été converti en masse de C et alloué à quatre catégories d'utilisation ultime des produits du bois : en usage, enfouissement, énergie et émission. La redistribution de C dans le temps parmi les catégories d'utilisation ultime des produits du bois a été calculée à l'aide d'une matrice de distribution de C basée sur l'âge des produits. Les émissions de C engendrées par la récolte, le transport et la transformation ainsi que la réduction des émissions due à l'utilisation du bois à la place d'autres matériaux de construction et des combustibles fossiles n'ont pas été comptabilisées. En tenant compte du bois récolté sur les terres publiques de l'Ontario de 1951 à 2000 et de la récolte prévue de 2001 à 2100, la séquestration de C dans les produits du bois en usage et dans les enfouissements devrait augmenter de 3,6 Mt-an⁻¹ de 2001 à 2100 et l'utilisation du bois à des fins énergétiques devrait générer 1,2 Mt-an⁻¹ de plus. Les prévisions d'ajout annuel de C dans les produits du bois dépassent largement l'augmentation annuelle de la séquestration de C dans les forêts publiques aménagées de l'Ontario qui devrait atteindre 0,1 Mt-an⁻¹ pendant la même période. Ces projections indiquent que la récolte réglementée en Ontario se traduit par un puits de C qui augmente régulièrement dans les produits du bois et les forêts. Les incertitudes associées à l'estimation de C dans les produits du bois sont également abordées dans la discussion.

[Traduit par la Rédaction]

Introduction

The default approach proposed by the Intergovernmental Panel on Climate Change (IPCC) for estimating and reporting carbon (C) stock changes in harvested wood products (HWP) is that the annual C inflow and outflow for the HWP reservoir are assumed to be equal, based on the perception that HWP stocks in most countries are not increasing significantly (IPCC 1997). However, Canada's submissions to the 20th Session of the Subsidiary Body for Scientific and Technological Advice (UNFCCC 2004) notes

general agreement with other countries' submissions that the default approach to HWP is inaccurate and may lead to policy decisions that will miss opportunities to reduce atmospheric greenhouse gas levels. Canada's statement is supported by a number of studies indicating that, each year, the amount of C stored in HWP increases at a rate of between one-quarter and one-half of the total annual biological C sequestration by the world's forests (Kellomäki and Karjalainen 1996; Nabuurs 1996). Winjum et al. (1998) and Brown et al. (1998) estimated that the global pool of C in HWP grows about 139–140 Mt-year⁻¹, whereas Pingoud et

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al. (2003) estimated this pool grows about 40 Mt·year⁻¹, excluding landfills. In the United States, the net increase of C in HWP in use and in landfills increased 1.7 times (from 22 to 59 Mt·year⁻¹) from 1970 to 1990 and was projected to increase to 75 Mt·year⁻¹ by 2040 (Skog and Nicholson 2000). Apps et al. (1999) estimated that for 1985–1989 the Canadian forest product sector produced a net increase of 23.5 Mt·year⁻¹ in stored C. These results all indicate that the amount of C stored in HWP is now large and continues to increase.

Quantifying the HWP C pool size and understanding the significance of forests and the forest sector on forest C storage and greenhouse gas emission reduction are important, especially among countries that signed the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol (Kurz et al. 1992; Nabuurs and Sikkema 2001; Birdsey and Lewis 2003; Heath and Skog 2004). Thus, the IPCC has been evaluating alternative approaches to accurately estimate and report C emissions and storage related to the life cycle of HWP (IPCC 1997, 2000, 2006; Brown et al. 1998; UNFCCC 2003).

Canada is one of the largest wood product producers and exporters in the world (UNFCCC 2003; FPAC 2006) and thus generates large amounts of C in HWP. Ontario's forests cover approximately 71.2×10^6 ha (17% of Canada's forest), of which 81% are Crown (i.e., public) forests (OMNR 2007). Thus, Ontario's HWP are critical in assessing the overall greenhouse gas impacts of Canada's forest sector. Despite their importance, little information is available on the C pool size and changes in HWP resulting from harvesting Ontario's Crown forests.

To address this shortcoming, this study (i) develops HWP C retention curves and a C-distribution matrix to describe the flow of C over time among end uses, (ii) estimates to the year 2100 the disposition of HWP C from historic harvesting (1951–2000), (iii) projects the disposition of HWP C from harvesting over the period 2001–2100, (iv) compares annual additions to C storage in HWP with annual additions to C storage in Ontario's managed Crown forests, and (v) assesses the effects of harvest on net C emissions or removals based on combined C storage in Ontario's Crown forests managed for harvest and the HWP from these forests.

Model description

Carbon accounting model

The IPCC (IPCC 1997, 2000, 2006; Brown et al. 1998) and UNFCCC (2003) have evaluated four approaches for estimating and reporting HWP C stock size and greenhouse gas emissions: (i) stock-change approach: changes in HWP C pools are estimated in HWP-consuming countries, regardless of where the HWP are produced, (ii) production approach: C changes in HWP pools are attributed to the HWP-producing countries, regardless of where the HWP are used, (iii) atmospheric-flow approach: net emissions or removals of C to or from the atmosphere, respectively, are estimated within each country's national boundaries, and (iv) simple decay approach: net emissions or removals of C

to or from the atmosphere, respectively, are estimated and reported when but not where they occur if HWP are traded.

The stock-change and production approaches focus on C stock changes in C pools, whereas the atmospheric-flow and simple decay approaches estimate and report gross C fluxes to or from the atmosphere (Cowie et al. 2006). The simple decay approach and the production approach should lead to the same national HWP C balance, because they both attribute HWP C changes to HWP producing countries. The stock-change and atmospheric-flow approaches should also provide similar estimates, because both estimate and report HWP C changes in HWP-consuming countries. All four methods will result in similar global HWP C stock estimates, so long as all data sources are complete, accurate, and consistent.

We used FORCARB-ON, an adaptation of the US national forest C budget model FORCARB2, to estimate C stored in HWP from wood harvested in Ontario (Colombo et al. 2007; Chen et al.³). Both FORCARB2 and FORCARB-ON use the production approach, in which HWP C is attributed to the area or region of harvesting (Heath et al. 1996; Birdsey and Lewis 2003). The production approach is recommended by IPCC (2003) for use when C storage changes associated with forest management within a certain land area are evaluated. Alternative approaches are more appropriate for evaluating the effect of factors that influence emissions from wood C (atmospheric-flow approach) or accumulation and loss of HWP C within national boundaries. Because our objective was to assess the effects of forest management in Ontario on global C cycle, alternative approaches are not discussed in this paper. The simple decay approach is similar to the production approach but is not implemented in FORCARB2 and FORCARB-ON, so it was not possible to report on it as part of this study.

FORCARB-ON converts harvested wood volume into C mass and then allocates the C mass into four HWP end-use categories: (1) HWP used for their primary intended purpose (in use), (2) HWP and processing residues disposed of in landfills (landfill), (3) HWP and processing residues burned to produce energy (energy), and (4) HWP and residues burned without producing energy or left to decompose (emission).

In our study, we used Ontario-specific data to improve the estimation accuracy of the provincial HWP stock. Allocation of HWP C to the four end-use categories was estimated by applying a C-distribution matrix, which describes the HWP C initial distribution over the four categories and subsequent C transfer from in use to other categories and from landfill to emission. Manufacturing and the associated C conversion efficiencies and energy production from burning harvested wood and processing residue determined the initial distribution of C over the four end-use categories (assuming all harvested wood is processed to HWP in the harvest year). Other factors, such as HWP lifetime in use, energy production from burning waste HWP, recycling, and HWP C decomposition in landfills, determined the transfer from in use to other categories and from landfill to emission.

³J. Chen, S.J. Colombo, M. Ter-Mikaelian, and L.S. Heath. 2008. Adapting the U.S. national forest carbon budget model to assess the forest carbon budget in Ontario, Canada. In review.

Fig. 1. Wood product C estimation in FORCARB-ON, an adaptation of FORCARB2 developed for use in Ontario. Wood product C is distributed among four categories: (i) in use, (ii) landfill, (iii) energy (burned to generate energy), and (iv) emissions (nonenergy use burning and decomposition).

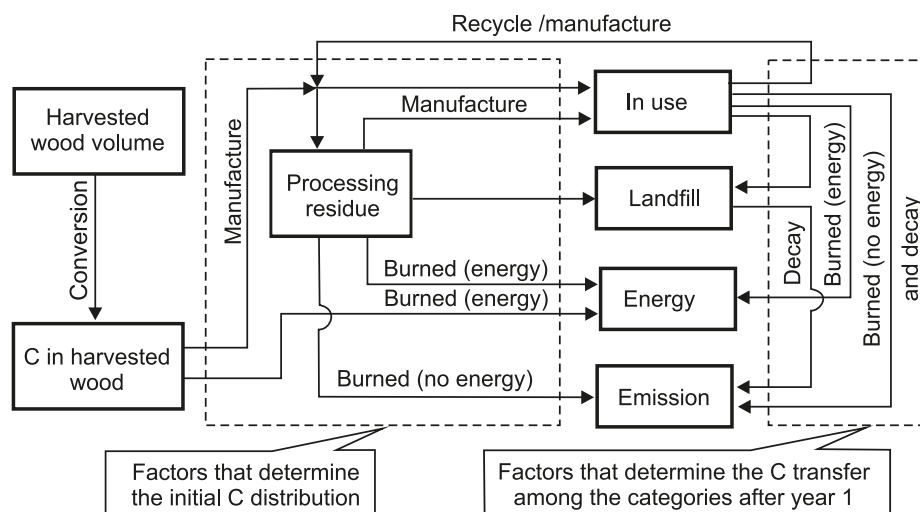


Table 1. Net merchantable wood volume (m^3 , $\times 1000$) harvested from Crown forests in Ontario between 1995 and 2004.

Year	Sawlogs and veneer		Composite panels		Pulpwood		Fuelwood		CF subtotal	HW subtotal	Total
	CF	HW	CF	HW	CF	HW	CF	HW			
1995	12 322	1115	2	1558	4408	1925	6	187	16 738	4785	21 523
1996	12 419	1203	10	2031	3546	1811	39	174	16 014	5219	21 233
1997	12 891	1447	15	1817	3816	1690	17 ^a	171	16 739	5125	21 864
1998	12 647	1335	12	2412	3151	1583	10	118	15 820	5448	21 268
1999	12 708	1163	16	2466	3952	2202	50	133 ^a	16 726	5949	22 675
2000	13 894	1445	15	2828	4655	2608	10	100	18 574	6981	25 555
2001	12 192	1248	15 ^a	3053	2275	1470	2	87	14 484	5858	20 342
2002	13 741	1225	25	2716	3934	2169	11	106	17 711	6216	23 927
2003	11 420	1084	24	2794	4203	2449	12	123	15 659	6451	22 109
2004	12 006	1222	28	2942	4159	2335	12	129	16 205	6628	22 833
Mean	12 624	1249	16	2462	3810	2024	17	133	16 467	5866	22 333

Note: Volume harvested is divided into primary harvested wood product categories and species group. CF, conifer species; HW, hardwood species. The table was reproduced from the National Forestry Database Program of Canada: www.nfdp.ccfm.org/compendium/data/2006_10/tables/com53e.pdf (accessed 13 September 2007).

^aThese three original values in the database were anomalous and were assumed to reflect errors in the database; they were replaced with mean values for the category.

Figure 1 presents the HWP C accounting model used in FORCARB-ON.

FORCARB-ON uses the harvested net merchantable volume estimates from a timber supply model as input data. In Ontario, Crown forests used for timber production are divided into 47 forest management units, and each unit has a forest management plan that is updated periodically. These forest management plans present estimates of present and future forest stand species composition, stand ages, and timber volume, modelled using the Strategic Forest Management Model (SFMM), Ontario's most commonly used timber supply model (Kloss 2002). We created an interface for FORCARB-ON to use SFMM harvest projections. The volume to C conversion factors in FORCARB2 were developed for different regions and forest types in the United States, and some of those designed for forests in the US Lake States Region (Birdsey 1992) were applied to similar forest types in Ontario. Using the same approach as in Kurz et al. (1992), we developed an Ontario-specific HWP

C-distribution matrix to partition HWP C among the four end-use categories over time. We also added a model component that allowed us to estimate HWP C stocks from historic harvesting. The modified model was used to produce a C budget for HWP from Ontario's Crown forests by decade from 2001 to 2100, including harvest from 1951 to 2000. The values are presented as inferred annual additions to the total C stocks in this paper.

Developing the C distribution matrix

Initial distribution of harvest wood volume among end-use categories

The initial distribution of C among primary HWP categories was considered the same as the distribution of wood volume because we assumed that each cubic metre of wood contained the same amount of C regardless of forest region or species.

We obtained Ontario's historic harvested volume data

Table 2. Distribution of harvested merchantable volumes (%) among primary wood product categories relative to the total harvested merchantable volume for conifers and hardwoods from Crown (i.e., public) forests in Ontario.

Wood product category	Harvested volume (%)		
	Conifer	Hardwood	Subtotal
Sawlogs, veneer logs, and composites	56.60	16.61	73.21
Pulpwood	17.06	9.06	26.12
Fuelwood	0.07	0.60	0.67
Total	73.73	26.27	100.00

Note: Values are calculated based on the mean harvest values for Ontario from 1995 to 2004 presented in Table 1.

Table 3. Conversion efficiencies for manufacturing primary wood products into secondary and end-use products (based on Kurz et al. 1992).

Primary products	Secondary or end-use product and residue	%
Sawlogs and veneer logs		
Softwood	Construction lumber	31.50
	Other lumber	19.00
	Pulp chips	24.75
	Processing residue	24.75
Hardwood	Other lumber	30.00
	Pulp chips	35.00
	Processing residue	35.00
Pulp logs		
Softwood and hardwood	Pulp chips	85.00
	Processing residue	15.00

(from 1951 onwards) from the National Forestry Database Program of Canada (www.nfdp.ccfm.org/compendium/data/2006_10/tables/com53e.pdf, accessed 13 September 2007). Because of lack of data, HWP C for harvesting prior to 1951 was not included in this study. Table 1 shows the amount of wood harvested from 1995 to 2004 from Crown forests in Ontario and its primary uses. Over this period, total harvested wood volume, the proportion of conifers and hardwoods, and the distribution of harvested wood among primary products varied little. Based on the mean annual merchantable volume distributed among primary HWP types, we calculated the ratio of each category to the total harvested merchantable volume (Table 2). These ratios were used to predict how future harvest volumes would be distributed among primary HWP categories. Primary HWP are processed to secondary and end-use HWP, producing residues (bark, sawdust, trims, chips, etc.). Harvested wood volume is reallocated among these HWP categories depending on processing methods, conversion efficiency, and the production and use of wood residues.

Tampier et al. (2004) estimated the annual merchantable volume of harvested wood in Canada to be 107.7 Mt-year⁻¹ dry mass, of which 89 Mt is sent to sawmills (34 Mt becomes pulp chips; 36 Mt, lumber; and 19 Mt, mill residue); 15 Mt is roundwood used for pulp; 2.1 Mt is used to produce poles, pilings, and composite board; and 1.6 Mt is fuelwood. Seventy percent of sawmill residue is used for other value-added products and energy, and 30% is disposed of in landfills or burned as waste. According to Hatton (1999), in western Canada on average 78% of a sawlog is converted

into an HWP: 40% as lumber and 38% as wood chips for pulp and paper. The remaining 22% becomes wood residue, including bark, sawdust, and shavings (also referred as hog fuel).

Kurz et al. (1992) analyzed wood-processing methods in Canada to estimate conversion efficiencies from primary to end-use HWP. Table 3 lists the conversion efficiencies from Kurz et al. (1992) used in the current study. Conversion efficiencies from chips and logs to pulp vary depending on pulping method and are not shown in this table.

Wood processing residue is used in secondary HWP, burned, or disposed of in landfills. The generation and use of wood residue must be considered when determining the initial HWP C distribution. In the United States, most solid wood residue is used as raw material for other processes or is burned for energy; only a small portion is left to decay or is burned as waste (Powell et al. 1993, table 36, p. 110). In Canada in the 1990s, wood residue use increased from 51% to 73% of its total production because of advances in mill recovery technologies and increased use of residues in value-added HWP and energy production (Hatton 1999). McCloy and Associates (1999) estimated that 70.6% of wood residue in Ontario was used in producing other HWP or generating energy, whereas Hatton (1999) estimated this utilization rate as 82.8%. We used the mean of these estimates, 76.7%, in our model simulations.

Figure 2 shows the initial C distribution among HWP categories and conversion from harvested wood volume to primary HWP and end-use HWP. This C distribution was produced using the information on harvesting and wood

Fig. 2. Harvested wood C distribution map for major wood product categories in Ontario based on the distribution of harvested wood volume over primary product types (Table 2) and the conversion efficiencies from primary products to secondary and end-use products (Kurz et al. 1992). Burning and decomposition are assumed to release 100% of affected C to the atmosphere.

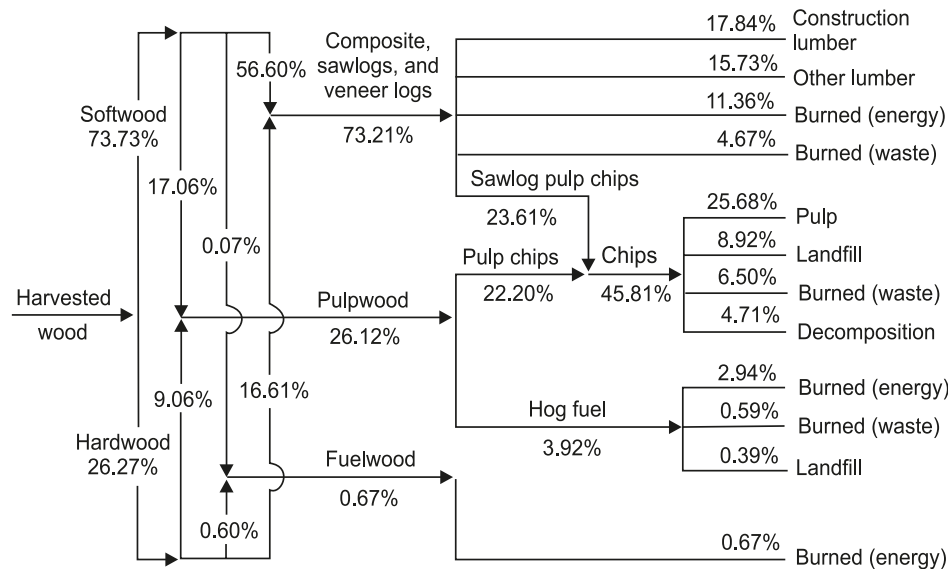


Table 4. Distribution of net merchantable wood volume harvested in Ontario among wood product end-use categories (based on Fig. 2).

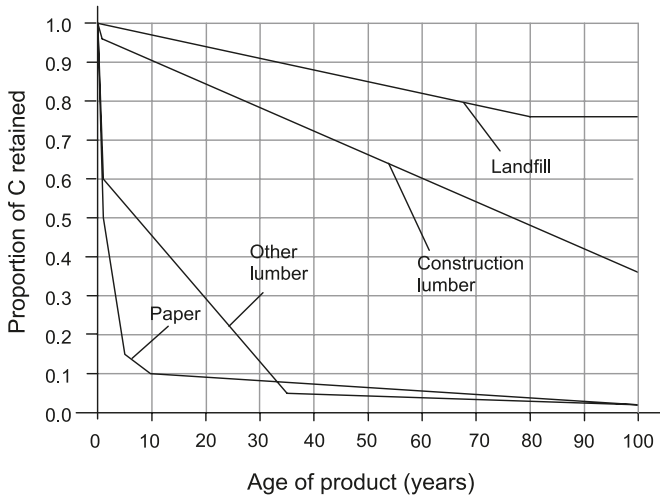
Category	Volume (%)
In use	
Construction lumber	17.84
Other lumber	15.73
Pulp	25.68
Total in use	59.25
Landfill	9.31
Burned to generate energy	14.97
Emission	
Burned as waste	11.76
Decomposition	4.71
Total emissions	16.47
Total	100.0

processing analyzed above, with C distribution assumed to be the same as wood volume distribution. Construction and other lumber accounted for 33.57% of the harvested volume; pulp, 25.68%; residue (bark, sawdust and shavings) and decomposition, 40.08%; and fuelwood, the remaining 0.67%. Table 4 summarizes the initial C distribution of harvested wood in the four end-use categories. This information was used to predict harvested wood C distribution in the year of harvest. Subsequently, estimates of C distribution in the four end-use categories were based on the fate of HWP.

Fate of HWP

Wood products have different service lives depending on their end use. After reaching the end of their service life, HWP are either recycled, burned (with or without energy generation), or discarded in landfills. Some C is lost before or when HWP are placed in their end uses. Kurz et al. (1992) assumed 5% construction lumber loss in the first year resulting from fitting and shaping. More recently, Skog

Fig. 3. Carbon retention curves for wood products in use and in landfills in Ontario. The retention curve for paper products is based on Kurz et al. (1992).



and Nicholson (2000) estimated 8% loss for solid HWP and 5% for paper and paperboard products as they are transferred to end uses.

Carbon also flows from in use to the other three end-use categories and from landfill to emission by decomposition. Wood products release C to the atmosphere slowly through decomposition and quickly and often completely through burning. Lumber remains in use for a long time, whereas most paper and paper products are disposed of more quickly. Half-life is the time it takes one-half of the C in a HWP to be removed from use (Skog and Nicholson 2000; UNFCCC 2003; IPCC 2006). According to Skog and Nicholson (2000), the half-life for major HWP end-use categories ranges from 67 to 100 years for construction lumber and from 1 to 6 years for paper and paper products. Others have presented similar half-life values (Kurz et al. 1992; Winjum

et al. 1998; UNFCCC 2003; Skog et al. 2004; US EPA 2007).

As shown in the C retention curve for wood in use (Fig. 3), we assumed a 5% C loss for construction lumber during the first year due to shaping and fitting (Kurz et al. 1992). For the remaining 95%, we used the mean half-life (79 years) for lumber in single-family homes, multifamily homes, and nonresidential construction (Skog and Nicholson 2000) as the overall half-life (Fig. 3). Carbon removal over time from this category was assumed to be linear.

Other lumber products were assumed to have 40% C removed from service in the first year, reflecting use in short-lived products such as pallets and packaging (Kurz et al. 1992). For the remaining 60%, we used the mean half-life (19.6 years) of mobile homes, pallets, furniture, and railroad ties estimated by Skog and Nicholson (2000). Thus, other lumber retains about 5% of its C at year 35, and we assumed it decreases to 2% by year 100 (Fig. 3).

For paper and paper products, Kurz et al. (1992) estimated 50% C loss in the first year resulting from the disposal of short-lived products, such as tissues and newsprint. This assumption was reflected in the C retention curve for paper (Fig. 3): 50% of C is retained in this HWP category by the end of year 1 declining to 15% at year 5 and to 10% at year 10. By year 100, 1% of C is predicted to remain in this HWP category, because it has few long-lived products (e.g., books and construction papers). As HWP C leaves the in-use category, it is reallocated to one of landfill, energy, or emissions, as per Kurz et al. (1992).

Recycling extends the lifetime of HWP, especially for paper products (Pingoud et al. 2003). According to FPAC (2006), paper recycling in Canada increased from about 28% in 1990 to 49% in 2006 and is expected to increase to 55% by 2010. For this study, extended HWP service lives due to recycling are taken into account in the C distribution matrix, especially for paper. Bioenergy production from all types of HWP has also increased (High and Skog 1990; Huber et al. 2005; Zerbe 2006), which was accounted for in allocating wood-processing residue and retired HWP into the energy-use category.

Harvested wood product C retention in landfills

Landfills store large stocks of HWP C. Wood and paper decay slowly in landfills (IPCC 2006), and improvements in technology and management are expected to further reduce C emission from HWP discarded in landfills (Micales and Skog 1997; Pingoud et al. 2003; UNFCCC 2003; IPCC 2006). Under anaerobic conditions that often exist in landfills, some wood compounds, such as lignin, decay very little (Micales and Skog 1997; Barlaz 1998; UNFCCC 2003).

In landfills, the proportion of degradable organic C that does degrade and is released to the atmosphere (DOC_f) and the portion that does not degrade ($1 - \text{DOC}_f$) vary depending on the type of HWP. Table 5 summarizes published long-term C storage factors for HWP in landfills. Earlier, the IPCC (1997) provided a default DOC_f value of 0.77, which was to be used where country-specific data (e.g., waste composition and waste decomposition rates in landfills) was unavailable. This default DOC_f value was based on a simple theoretical equation relating landfill temperature (T , assumed constant at 35 °C) to organic matter decomposi-

Table 5. Published long-term C storage factors representing the proportions of the original wood and paper products that are very slow to decompose in landfills.

Methods to obtain data or original data sources	Coated paper	Newsprint	Corrugated box	Office paper	Wood, bark, and branch	Reference
Theoretical model						IPCC 1997
Literature review						IPCC 2006
IPCC (1997) default values			0.23 ^a			Environment Canada 2006
IPCC (1997) methodology with estimated landfill temperature (30 °C)	0.23 ^a (MSW); 0.5 (WW) ^b		0.5 ^a			Swedish EPA 2004
IPCC (1997) default value			0.5 ^a			Statistics Finland 2006
IPCC default values adjusted according to Finnish conditions		0.53			0.57	Pipatti 2001
Laboratory experiments based on Doorn and Barlaz (1995)	0.79	0.84	0.61			Eleazer et al. 1997
Laboratory experiments based on Doorn and Barlaz (1995)	0.75	0.85	0.55	0.125	0.77	Barlaz 1998
Based on Barlaz (1998)	0.75	0.85	0.55	0.12	0.77	US EPA 2007
Based on Barlaz (1998)					0.77	Smith et al. 2006
Analyzed samples retrieved from landfills					0.965–0.986	Ximenes et al. 2005
Based on US EPA (2002)	0.85	0.93	0.58	0.11		Miner 2003
Recalculated from Doorn and Barlaz (1995)	0.825	0.843	0.685	0.618	0.968–1.000	Micales and Skog 1997
		0.738 ^c				

^aFor all degradable organic C in landfills.

^bMSW, municipal solid waste landfill; WW, wood waste landfill.

^cMean value for all paper products from Micales and Skog (1997).

Table 6. Carbon distribution matrix for harvested wood products (HWP) from Ontario.

Category	HWP age (years)										
	10	20	30	40	50	60	70	80	90	100	110
In use	0.347	0.306	0.271	0.235	0.223	0.210	0.197	0.185	0.172	0.160	0.153
Landfill	0.308	0.333	0.353	0.373	0.373	0.373	0.372	0.372	0.372	0.372	0.372
Energy	0.166	0.170	0.172	0.175	0.176	0.177	0.178	0.179	0.180	0.181	0.181
Emission	0.179	0.192	0.204	0.217	0.229	0.241	0.253	0.264	0.276	0.288	0.294

Note: Values are the proportional distribution of C in wood products among four categories: (1) in use, (2) landfill, (3) energy (burned to generate energy), and (4) emission (from nonenergy use burning and decomposition). Proportions in each year add up to 1.

tion, in which $\text{DOC}_f = 0.014T + 0.28$. This earlier default value did not consider important factors affecting DOC_f , such as the type of organic waste in the landfill, moisture, and pH. Based on a literature review, the IPCC concluded that this default DOC_f value might overestimate decomposition of degradable organic C in landfills, so the default DOC_f was changed to 0.5–0.6 (IPCC 2000) and then to 0.5 (IPCC 2006).

The most recent default DOC_f value (IPCC 2006) of 0.5 applies to all solid organic waste (not just HWP containing lignin). Therefore, smaller DOC_f values (i.e., greater long-term C storage factors) should reasonably apply to HWP and wood processing residue that are disposed of in landfills. Carbon storage factors reported in Eleazer et al. (1997) and Barlaz (1998) were based on a series of experiments using laboratory-scale landfills and considering the major factors that affect the decomposition of organic matter therein; the resulting C storage factors were used to estimate US national greenhouse gas emissions and sinks (US EPA 2007). In their experiments, ideal anaerobic decomposition conditions were provided to laboratory landfills, maximizing methane production (Eleazer et al. 1997, Barlaz 1998). However, landfills often do not have ideal conditions, and these storage factors might overestimate HWP C decomposition. Therefore, in this study, we used C storage factors from Barlaz (1998), which we consider appropriate to avoid overestimating HWP C storage in landfills (Table 5).

We estimated that 53% of HWP C entering landfills is from discarded solid wood products and 47% is paper waste from harvested Ontario forests. These estimates were based on species harvested and product types shown in Table 1, conversion efficiency in HWP production (Kurz et al. 1992), and HWP recycling rates (FPAC 2006). Proportions of paper product types in landfills were taken from EPIC (2002), which estimated that newsprint, office paper, corrugated cardboard, and mixed paper account for 41%, 9%, 12%, and 38%, respectively, of municipal waste in Canada.

A weighted mean long-term C storage factor (73.1%) was produced based on the HWP waste composition and the C storage factors reported in Barlaz (1998). Using this C storage factor, we assumed that 73.1% of HWP C remains in landfills indefinitely, and the remaining 26.9% decomposes linearly over 80 years (Fig. 3).

Using the initial HWP C distribution given in Table 4 and the C retention curves in Fig. 3, a C redistribution matrix for HWP was developed (Table 6). Using the percentages given in this matrix, HWP C was distributed among the four end-use categories based on HWP age.

Results and discussion

Future C stocks and changes in HWP from harvesting Ontario's Crown forests were compiled from estimates for each of the 47 provincial forest management units. Our estimates of HWP C start in 2001 (the first year of the current 5 year forest management plans for most management units) and cover a simulation period of 100 years. We did not include C emissions from harvest, manufacturing, and transport of HWP; avoided emissions due to the use of wood in place of fossil fuels; or reduced emissions due to the use of HWP in place of fossil-fuel-intensive building materials.

Ontario's SFMM model produces a maximum available future harvest area. When we analyzed the actual harvest data from 1991 to 1998, we found that actual harvest area averaged only 66% of the available harvest area in Ontario's Northwestern and Northeastern regions and only 54% in the Southern Region. Because only 5 of the 47 management units are in the Southern Region, we used a mean harvest rate of 65% to predict future harvest area for all Ontario management units.

Future HWP C storage projected from historic and future forest harvesting in Ontario Crown forests is shown in Tables 7 and 8. Harvest from 1951 to 2000 and predicted harvest from 2001 to 2100 was projected to move 871.5 Mt C in wood from Ontario Crown forests into the HWP stream. By year 2100, 490.2 Mt C of that total (or 56.3%) was projected to be stored in HWP in use and in landfills, and 222.0 Mt C (25.5%) was projected to be released to the atmosphere by decomposition or burning without producing energy. Energy production was estimated to consume 159.3 Mt (or 18.3%) of the total HWP C, mainly from burning wood processing residue to generate energy in the forest products industry. Considering the disposition of all harvest from 1951 to 2100, the C storage in HWP in use and in landfills was projected to increase by a mean of 3.6 Mt-year⁻¹ over the period 2001–2100, with an additional 1.2 Mt C-year⁻¹ burned for energy production. Carbon in HWP in use was projected to increase by 1.3 Mt-year⁻¹, whereas C in landfills was projected to increase at about 2.3 Mt-year⁻¹.

Using FORCARB-ON, we also projected that C storage in Crown forests managed for harvest will increase from 4278.0 Mt in 2001 to 4291.5 Mt in 2100, a mean increase of 0.1 Mt-year⁻¹ during this century (Colombo et al. 2007). We note that there are additional areas of managed Crown forests in Ontario that are not subject to harvest. The total managed Crown forest C sink in Ontario is on the order of 0.7 Mt-year⁻¹ (Colombo et al. 2007). Thus, projected C stor-

Table 7. Wood product C storage and emissions (Mt) for projected harvesting by decade from 2001 to 2100.

Category	Year										
	2001	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
In use	0.0	25.2	45.9	62.7	76.8	90.2	103.3	115.6	128.3	140.6	152.3
Landfill	0.0	22.2	45.0	67.8	91.4	114.7	138.1	161.5	185.9	210.9	236.4
Energy	0.0	12.0	23.5	34.6	45.4	56.3	67.4	78.5	90.3	102.4	114.8
Emission	0.0	13.0	26.0	39.2	53.0	67.4	82.6	98.7	116.0	134.5	154.0

Note: Carbon is distributed among four categories: (1) in use, (2) landfill, (3) energy (burned to generate energy), and (4) emission (from nonenergy use burning and decomposition).

Table 8. Wood products carbon storage and emissions (Mt) by decade from historic harvesting from 1951 to 2100.

Category	Year															
	1951	1960	1970	1980	1990	2000	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
In use	0.0	8.8	15.2	22.0	32.5	44.2	39.2	35.1	32.0	30.0	28.3	26.6	25.0	23.5	21.9	20.7
Landfill	0.0	10.7	24.1	39.4	61.2	81.6	83.0	83.4	83.1	82.0	81.1	80.5	80.4	80.5	80.6	80.7
Energy	0.0	6.5	13.5	21.6	33.3	42.8	43.1	43.4	43.6	43.8	43.9	44.0	44.2	44.3	44.4	44.5
Emission	0.0	5.9	12.8	21.0	33.2	45.5	48.8	52.1	55.3	58.2	60.8	62.9	64.5	65.8	67.1	68.0

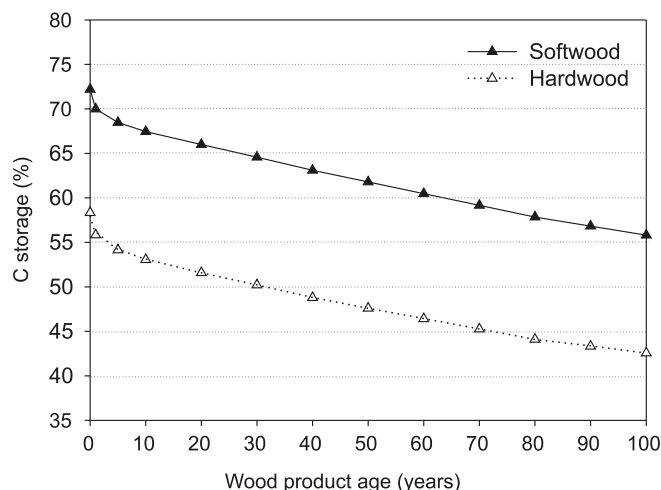
Note: Carbon is distributed among four categories: (1) in use, (2) landfill, (3) energy (burned to generate energy), and (4) emission (from nonenergy use burning and decomposition).

age in HWP far outweighs the C storage increase projected for all of Ontario's managed Crown forests. Therefore, HWP C has significant implications when assessing the effects of harvesting on total forest sector C storage.

Using the harvested wood C distribution map (Fig. 2), the C retention curves (Fig. 3), and other parameters described in this report, we separately estimated the proportions of C stored in softwood and hardwood HWP (including HWP in use and in landfills) (Fig. 4). In the year of harvesting (assuming harvested wood is processed into HWP in the same year), 72.2% of C of the softwood removed by harvest was projected to be stored in HWP in use and in landfills, which was projected to decrease to 55.8% by year 100. For hardwoods, it was projected to decrease from 58.3% in the year of harvesting to 42.6% by year 100. The higher percentage of C retained in softwood products is mainly due to the use of softwoods as construction and other lumber, whereas hardwoods are more often used to make shorter lived HWP (Fig. 2; Table 3). The rapid C storage decrease in the two curves in the first few years is due to burning wood and processing residue as a biofuel or burning short-lived HWP. Decomposition is less important and occurs over the whole HWP lifetime, continuing after they are discarded in landfills.

Given the large volumes harvested from Ontario's Crown forests, the use of wood for different types of products, and their consumption in a variety of jurisdictions and end-users, a modelling approach seems the most practical way to estimate the HWP C stock changes. However, some important parameters used in the models are based on estimates that are difficult to verify by direct measurement. Modelling also relies on assumptions to simplify complicated processes that occur in nature, e.g., that HWP leave the in-use category at a constant rate, and the decomposition of HWP in landfills is a constant fraction of the total HWP in landfills. Such simplified assumptions increase the uncertainties in estimating HWP C storage and emissions.

Our simulation showed that HWP and wood residue in landfills are an important C stock, which increases rapidly

Fig. 4. Carbon retention curves for softwood and hardwood products in use and in landfills combined in Ontario.

compared with HWP in use. Accurate prediction of DOC_f values and long-term C storage factors for HWP in landfills are needed to correctly estimate the size of this C storage. In comparison with the values used in this study, Pipatti (2001) and the Swedish EPA (2004) used the earlier IPCC default DOC_f value (IPCC 1997); therefore, they likely overestimated solid waste decomposition in landfills and, thus, underestimated C stock. Environment Canada (2006) selected 0.77 and 0.5 as the default DOC_f for municipal solid waste and wood waste in landfills, respectively, based on past default values used by the IPCC (1997). More recently, the IPCC (2000) recognized that the default DOC_f value of 0.77 should be used only if lignin-containing organic material is excluded. Municipal solid waste normally includes a large amount of lignin C in discarded solid wood and paper products. Thus, Environment Canada (2006) likely also underestimated the size of the HWP C stock in landfills.

Micales and Skog (1997) presented C storage factors for various types of woody materials in landfills based on val-

ues reported in Doorn and Barlaz (1995). These C storage factors have been used in FORCARB2 (Woodbury et al. 2007). However, the wood and paper decomposition data presented in Doorn and Barlaz (1995) were later updated by Barlaz (1998), who reported lower C storage factors for office paper and solid wood in landfills. Therefore, Micales and Skog (1997) likely overestimated HWP C in landfills.

Parameters describing long-term C storage of HWP in landfills used in this report were derived from studies based on laboratory landfill experiments (Eleazer et al. 1997; Barlaz 1998). Because conditions in real landfills are complicated and may differ from one to another, it is critical to verify landfill C storage factors for accurate assessment of HWP stock size. This could be done by excavating HWP samples from different landfills located in different areas in Canada and the United States and measuring the C content and analyzing C decomposition (and indirectly, emissions) of the samples. Unfortunately, this information is costly and time-consuming to obtain; as a result, very few such studies have been conducted and published. This increases uncertainty in HWP C storage or emission estimation in landfills, because a small error in a long-term C storage factor would cumulatively result in large differences in estimated landfill HWP C storage, either larger or smaller than those estimated in this study.

We believe it is reasonable to assume that modern landfills in Canada and the United States are designed and operated similarly, thus obviating the need to use different factors for each country. In the latest US greenhouse gas emissions and sinks inventory report (US EPA 2007), data from studies by Eleazer et al. (1997) and Barlaz (1998) were used in producing long-term HWP C storage factors in landfills, which were considered appropriate to apply to Ontario landfills.

Another important parameter in estimating HWP C storage is product half-life. Skog et al. (2004) concluded that this parameter may not be of key importance in total HWP C estimation. An estimated half-life greater than the actual one will delay the C conversion of HWP C from in-use to energy, emission, and (mostly) landfill. Regardless of earlier or later conversion, most of the HWP C either remains in the in-use category or moves to the landfill category. The error in total HWP C storage resulting from an error in half-life consists of two components: (i) a small portion of the retired HWP is burned and the C is emitted earlier or later than would actually occur, and (ii) the decay rates of HWP in in-use and landfill categories might be different, generating an error in decomposition for the affected HWP.

The half-life values used in the latest US greenhouse gas inventory report (US EPA 2007) and other US studies on HWP C storage (Winjum et al. 1998; Skog and Nicholson 2000; Woodbury et al. 2007) are similar to values used in this study. For example, the half-life of a single-family house built in the 1920s in the United States is 80 years, which is confirmed by US census data (US EPA 2007). In this study, we used 80 years as the half-life for single-family houses built before 1980, based on Skog and Nicholson (2000).

Advances in wood processing technologies will likely result in higher HWP conversion efficiencies and increases in wood residue utilization and may increase HWP life. The

trend towards increased recycling will further prolong the life of HWP. As a result, our projections likely underestimate future HWP C values in the in-use and landfill categories for the same projected harvest. Increasing demand for bioenergy will also increase the proportion of HWP C used to produce energy. Thus, we consider our projections of future long-term C storage in HWP and bioenergy production from Ontario's Crown forests to be conservative estimates.

In addition to forest and HWP C, a more complete analysis of the net C balance requires consideration of: (i) greenhouse gas emissions from the harvest and transport of logs and manufacturing and transport of products to markets and (ii) reduced greenhouse gas emissions from the use of wood in place of fossil fuels and energy-intensive building materials. In a future analysis of HWP C for Ontario, we aim to refine these estimates by accounting for these factors.

In addition, C trade-offs between disposal of HWP in landfills on the one hand and energy generation using HWP waste to displace fossil fuels on the other need to be considered. Even though decomposition of HWP in landfills is slow, emissions may still total almost one-quarter of the total HWP C placed there. As a result, disposal of HWP waste in landfills still releases a large amount of C through decomposition. In comparison, using HWP to generate energy is considered C neutral when it replaces fossil fuels, avoiding those emissions. Thus, using HWP at the end of their service lives for energy generation will generally reduce net emissions more than placing HWP in landfills.

Landfill gas recovery has become more common as a measure to reduce methane emissions. This is important, because methane has a global warming potential 25 times greater than CO₂ over 100 years (IPCC 2007) and accounts for about 50% of greenhouse gas produced from landfills (IPCC 2006). In 2004, 21% of methane produced from landfills in Canada was captured for energy recovery purposes, and the amount captured from landfills is expected to grow quickly (Environment Canada 2006). This recovery is not accounted for in FORCARB-ON, and thus, changes in greenhouse gas emissions due to landfill gas recovery and energy generation were not accounted for in our present estimates. To improve their accuracy, future estimates will need to account for methane capture and its use for energy generation.

Exports of HWP produced in Canada accounted for 79% of all production in 1999 (CCFM 2006) and US markets accounted for more than 94% of Ontario's HWP exports between 1990 and 2006 (canadaforests.nrcan.gc.ca/statsprofile/, accessed 8 Feb. 2008). We applied these export factors to HWP from Ontario (i.e., 79% exported, of which 94% go to the United States), and calculated that the Canadian domestic market and US markets were consuming 96% Ontario's HWP. Primary HWP (e.g., logs and bolts, pulpwood, and wood chips) account for only 0.5%–1.6% of Ontario's exported HWP, whereas the remainder are end-use HWP manufactured in Ontario (pulp and paper products, and wood-fabricated materials such as lumber, oriented strandboard, plywood, and veneer; canadaforests.nrcan.gc.ca/statsprofile/, accessed 8 Feb. 2008). We assume that conversion efficiencies for exported primary HWP in the United States would be similar if not higher than in Canada and,

thus, would not affect the initial C distribution among the four HWP end-use categories.

Our analysis does not account for future changes in the relative amounts of primary, secondary, and end-use HWP. In addition, the lifetimes of HWP in use may change in the future, which would also change the distribution of C among HWP categories and the HWP C storage and emission factors. One such change could be an increase in the proportion of harvest of hardwoods compared with softwoods, because much of the available merchantable hardwood volumes are not presently harvested.

Of Ontario's 71.2×10^6 ha of forests, private lands are 11% of the area but provide approximately 17% of the harvested wood (OMNR 2007). Our analysis does not include private land forests or their wood products. However, the relatively higher harvest volume from private forest lands may mean they have relatively lower forest C and higher HWP C compared with managed Crown forests.

This analysis of Ontario Crown forest net C storage in HWP and forests does not account for emissions from HWP placed in use prior to 1951. Estimating the present size of the pre-1951 HWP stock still in use is problematic, but it is likely much smaller than the size of the 1951–2000 HWP stock. The 1900–1950 harvest was likely less than the subsequent 50 year period, because the populations of the northeastern United States and Canada have grown substantially since 1950, and it is reasonable to assume a direct correlation between population and wood use. In addition, it is probable that a smaller proportion of the pre-1951 harvest was used in long-lived HWP (e.g., housing) than is presently the case, and conversion efficiencies prior to 1951 were likely lower. It is also probable that wood retired from service prior to 1951 as well as processing residues were less likely to be sent to landfills and more likely to be burned or left aboveground, leading to more rapid decay and higher short-term emissions. For the above reasons, we anticipate that HWP created before 1951 would be far less substantial than the post-1951 HWP stock.

Thus, an upper bound of annual C emission over the period 2001–2100 for HWP placed in use from 1901 to 1950 can be estimated as the annual C emission of HWP placed in use from 1951 to 2000 over a comparable period 2051–2150. The FORCARB-ON projections we conducted did not provide values for C emissions beyond 2100 for HWP placed in use from 1951 to 2000. However, these emissions would be less than those during 2051–2100, because emissions from HWP in use and in landfills decline with HWP age, as shown in Tables 6 and 7. Thus, C emissions over the period 2001–2100 for HWP placed in use from 1901 to 1950 should be less than the doubled emission of HWP placed in use from 1951 to 2000 over the period 2051 to 2100. Using C emission data for HWP from historic harvest (Table 8), we calculated an upper bound for these C emissions as $0.14 \text{ Mt} \cdot \text{year}^{-1}$. Consequently, we conclude that C emissions from HWP placed in use prior to 1951 would have only a minor effect on the net C storage or emission balance for HWP from Ontario's Crown forests over the period 2001 to 2100.

Forest harvesting in Ontario, as practiced within the context of its legislated need for sustainable forest management and taking into account HWP C is projected to result in

large long-term increases in C storage and atmospheric greenhouse gas reductions (Colombo et al. 2005, 2007). Forest fire burns forest floor, down woody debris, and standing trees, immediately releasing a large amount of C to the atmosphere, followed by more gradual C emissions as wood in fire-killed trees decomposes. However, fire protection in Ontario's managed forests reduces the total burned area and the associated emissions (Ward et al. 2001). The sustainable harvest level in Ontario is based on this reduction in area burned. Sustainable management also requires prompt regeneration of harvested forests. With either wildfire or harvest, forests regenerating after disturbance will grow and sequester C, recapturing C lost because of decomposition of branches, foliage, and root systems. However, in burned stands, the trees will decompose, and the C they contain will be released to the atmosphere, whereas much of the merchantable C in HWP from harvested stands remains in use or in landfills beyond 100 years after disturbance. This creates a more favourable overall C balance for managed than unmanaged forests.

In conclusion, this study shows that projected C storage in HWP from Ontario's Crown forests, based on projected forest harvesting for 2001–2100 and historic harvesting for 1951–2000, will create a steadily increasing C sink in HWP and forests between 2001 and 2100. Ontario Crown forests managed for harvest based on principles of sustainability are projected to create a mean annual C sink of 0.1 Mt. Including HWP, the combined C storage is projected to grow on average by 3.7 Mt annually. These estimates do not include emissions related to harvest, transport, or manufacturing nor do they include reduced emissions from using wood in place of building materials that are more energy intensive to produce and use nor avoided emissions when using wood in place of fossil fuels. We intend to report the combined effects of these factors on C stocks of HWP from Ontario's Crown forests in future analyses.

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